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EFFECT OF WATERSHED SUBDIVISION AND FILTER WIDTH ON SWAT SIMULATION OF A COASTAL PLAIN WATERSHED[1]

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ABSTRACT: The Soil and Water Assessment Tool (SWAT) does not fully simulate riparian buffers, but has a simple filter function that is responsive to filter strip width (FILTERW). The objectives of this study were to (1) evaluate SWAT hydrology and water quality response to changes in watershed subdivision levels and different FILTERW configurations and (2) provide guidance for selecting appropriate watershed subdivision for model runs that include the riparian buffer feature through the FILTERW parameter. Watershed subdivision level is controlled by the critical source area (CSA) which defines the minimum drainage area required to form the origin of a stream. SWAT was calibrated on a 15.7 km^2 subdrainage within the Little River Experimental Watershed, Georgia. The calibrated parameter set was applied to 32 watershed configurations consisting of four FILTERW representations for each of eight CSA levels. Streamflow predictions were stable regardless of watershed subdivision and FILTERW configuration. Predicted sediment and nutrient loads from upland areas decreased as CSA increased when spatial variations of riparian buffers are considered. Sediment and nutrient yield at the watershed outlet was responsive to different combinations of CSA and FILTERW depending on selected in-stream processes. CSA ranges which provide stable sediment and nutrient yields at the watershed outlet was suggested for avoiding significant modifications in selected parameter set.

(KEY TERMS: SWAT; riparian buffer; nonpoint source pollution; nutrients; sediment; simulation; watershed.)

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INTRODUCTION

Riparian forest buffers (RFBs) have been shown to be effective at reducing sediment and nutrient transport from agricultural lands to streams (Lowrance *et al.*, 1984; Correll, 2005; Lowrance and Sheridan, 2005). Lowrance *et al.* (1983) noted effective removal of nitrogen and transformation of nitrogen form from NO_3-N to organic-N by the RFB in an experimental watershed in the Coastal Plain of Georgia. Several modeling approaches have been developed to evaluate the effectiveness of RFBs on reducing pollutant transport to streams from upland agricultural areas

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(Inamdar *et al.*, 1999; Lowrance *et al.*, 2000; Gerwig *et al.*, 2001; Cerucci and Conrad, 2003; Dukes and Evans, 2003). However, most of these studies to quantify the water quality benefits of riparian buffers under varying site conditions are based on field-scale data and field-scale applications of models such as the Riparian Ecosystem Management Model (REMM). Dukes and Evans (2003) evaluated and tested the sensitivity of the hydrology component of REMM by comparing daily simulated water table depth to the observed water table depth at a field in the North Carolina Middle Coastal Plain. Gerwig *et al.* (2001) used Groundwater Loading Effects of Agricultural Management Systems and REMM to estimate nitrogen and phosphorus transport through a riparian buffer zone from an agricultural field that received swine lagoon effluent. Inamdar *et al.* (1999) used REMM to simulate nitrogen, phosphorus, and carbon cycling and transport in a Coastal Plain riparian buffer system near Tifton, Georgia. Lowrance *et al.* (2000) also used REMM for simulating groundwater nitrate concentrations and water table depths in a mature RFB in a Coastal Plain, Georgia. Watershed-scale modeling studies on the effects of RFBs are rare. In one such study, Cerucci and Conrad (2003) applied SWAT and REMM in 37 km^2 of watershed in Delaware County, New York to predict the loads from different source areas with and without riparian buffers for selecting the most cost efficient parcels to form a riparian buffer.

The Soil and Water Assessment Tool (SWAT) is a physically based, semi-distributed watershed-scale model that has been used extensively to predict the impact of land management practices on water, sediment, and agricultural chemical transport (Gassman *et al.*, 2007). SWAT, however, is limited by the way it represents streamside RFBs, resulting in a simplification of transport paths of sediment and nutrients from upland fields to receiving waters. Surface and subsurface trapping efficiencies are calculated within SWAT using the width of filter strips at the edge of the field (FILTERW). FILTERW is a user input for each hydrologic response unit (HRU) and can be used to estimate sediment and nutrient removal by RFBs (Neitsch *et al.*, 2005). As a result, simulated sediment yields and nutrient yields at the watershed outlet are sensitive to FILTERW selection. Appropriate representation of the RFBs is important for southeastern Coastal Plain watersheds where dense dendritic networks of stream channels are commonly bordered by RFBs.

The spatial distribution and density of RFBs can be derived using a geographic information system (GIS). Computational units within SWAT, including subwatersheds and HRUs, are also generally derived using a GIS interface (Di Luzio *et al.*, 2005). However, the number and size of delineated subwatersheds and density of channel networks are sensitive to a user-defined critical source area (CSA). The CSA is a user input within the GIS interface defining the minimum drainage area required to form the origin of a stream (Winchell *et al.*, 2007). Previous studies of SWAT responsiveness to different CSA levels indicate that sediment and nutrient simulations are sensitive to the number and size of subwatersheds (FitzHugh and Mackay, 2000; Jha *et al.*, 2004; Arabi *et al.*, 2006). None of these previous studies, however, considered the spatial characterization of RFBs in evaluating SWAT sensitivity to different watershed configurations.

The objectives of this study are to (1) evaluate SWAT hydrology and water quality response to changes in watershed subdivision levels and different FILTERW configurations and (2) provide guidance for selecting appropriate watershed subdivision for model runs that include the riparian buffer feature through the FILTERW parameter.

STUDY AREA AND AVAILABLE DATA

The Little River Experimental Watershed (LREW) is located near Tifton, Georgia, in the Gulf-Atlantic Coastal Plain (Figure 1). LREW is one of 14 USDA-Agricultural Research Service (ARS) benchmark watersheds used by ARS as part of the Conservation Effects Assessment Project (USDA, 2007). The soils, land use, topography, geology, and conservation

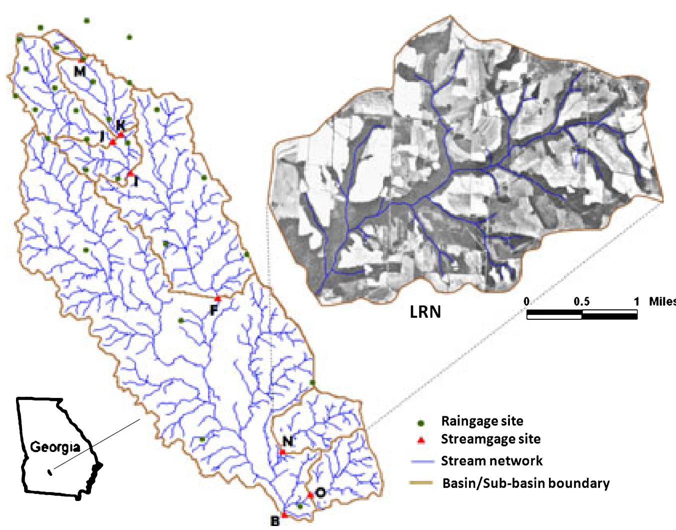

FIGURE 1. Location of Little River Experimental Watershed and Monitoring Networks.

practices within LREW are representative of much of the middle Gulf-Atlantic Coastal Plain of Georgia. The region has low topographic relief and is characterized by broad, flat alluvial floodplains, river terraces, and gently sloping uplands (Sheridan, 1997). The soils are underlain by a plinthic layer of lower permeability at 0.9-1.5 m. The LREW landscape is dominated by a dense dendritic network of stream channels bordered by riparian forests (Figure 1). These riparian areas provide storage for storm runoff and lateral groundwater flow from adjacent upland areas and have great potential for buffering the impacts of runoff from agricultural areas (Asmussen *et al.*, 1979). Field and watershed-scale functions of RFBs and combined forest and grass buffers have been previously documented at LREW (Yates and Sheridan, 1983; Lowrance *et al.*, 1984; Lowrance and Sheridan, 2005). Hydrology and water quality at LREW have been monitored by the ARS Southeast Watershed Research Laboratory (SEWRL) since the 1960s (Bosch *et al.*, 2007; Feyereisen *et al.*, 2007).

The 15.7 km^2 Little River subwatershed N (LRN) was selected for this study. Major soil series in LRN include Tifton loamy sand (48%), Alapaha loamy sand (16%), and Kinston and Osier fine sandy loam (6%). Tifton series soils covering most of the agricultural land have moderate infiltration rates (hydrologic soil group B), while Alapaha and Kinston-Osier soils are located around the stream and wetland areas and are in hydrologic soil group D. Row crop, pasture, and riparian forest areas cover approximately 41, 13, and 30% of LRN, respectively. The remaining 16% of LRN includes roads, residences, fallow land, and other land uses (Lowrance *et al.*, 1984).

The AVSWAT-X user interface was used for creating model input from readily available GIS data. A land cover map derived from 1998 Landsat imagery, with a spatial resolution of 30 m, was downloaded from the Georgia GIS Clearinghouse (https://gis1.state.ga.us/). Detailed land cover categories were reclassified into 12 SWAT land use codes. Soil Survey Geographic (SSURGO) data from the USDA-NRCS (http://soildatamart.nrcs.usda.gov/) were used. A digital elevation model with 30 m grid resolution obtained from USGS (http://seamless.usgs.gov/website/seamless/) was used to delineate subwatershed boundaries and stream networks according to different CSAs.

SWAT DESCRIPTION

The PC version of SWAT2005 was used to model hydrology and water quality response to different combinations of RFB representations and watershed subdivision levels. In AVSWAT-X, a watershed is divided into multiple subwatersheds based on a user defined threshold area (CSA) and the subwatersheds are then further subdivided into HRUs, the smallest calculation unit in the model. Calculated overland flow and pollutant loads from HRUs within a subwatershed are lumped into the corresponding stream without interactions between HRUs within the subwatershed. Collected water and pollutant loads from each subwatershed are then transported to the watershed outlet by considering interactions between channel segments (Neitsch *et al.*, 2005).

Daily surface runoff volume from each HRU is calculated using the SCS curve number (CN) method. The variable storage routing method or Muskingum routing method can be used for computing water transport through stream networks (Neitsch *et al.*, 2005). Sediment erosion from upland areas and delivery to the stream is estimated using the Modified Universal Soil Loss Equation (MUSLE) (Williams, 1975). MUSLE estimates sediment yield for a single rainfall event by replacing the rainfall erosivity index in USLE (Wischmeier and Smith, 1978) with runoff factors, such as the surface runoff volume and peak runoff rate. Other USLE factors such as soil erodibility, slope-length, slope-steepness, cover and management, and erosion-control practice factors are still used in MUSLE. Sediment routing in streams is estimated by comparing available sediment loads and estimated sediment transport capacity within each stream segment. If the transport capacity is greater than available sediment load within a given stream segment, channel degradation occurs. If transport capacity is less than available sediment load, the sediment will be deposited within the stream.

The transformation and movement of nutrients within an HRU are simulated based on several soil inorganic and organic pools. The respective amounts of nitrate (NO_3-N) contained in runoff, lateral flow, and percolation, are estimated as products of the water volume and the average concentration of nitrate in each soil layer. The amount of soluble P removed by runoff is predicted as a function of solution P concentration in the top 10 mm of soil, the runoff volume, and a partitioning factor. Organic N and phosphorus transport with sediment are calculated from loading function developed by McElroy *et al.* (1976) and modified by Williams and Hann (1978) for application to individual rainfall runoff events. The loading function estimates the daily nutrient loss based on the concentration of nutrient in the top soil layer, the sediment yield, and an enrichment ratio. The loss of soil nutrients through plant uptake is also estimated. In-stream nutrient transformations are optionally considered only when in-stream kinetics adapted from the QUAL2E model are selected. Nutrients dissolved in the stream are transported with the water while those adsorbed to sediments are allowed to be deposited with the sediment on the bed of the channel. Nutrient cycling in each stream segment is described as a function of travel time and transformation rates among different nutrient forms: organic, ammonium, nitrite, and nitrate forms for nitrogen; and organic and soluble forms for phosphorus. Growth and death of algae, an additional component of in-stream nutrient cycling, is calculated as a function of growth rate, respiration rate, and settling rate factors. The growth rate of algae is involved in the in-stream nutrient cycles through the growth limitation factor for nutrient and the limitation factor is internally estimated based on nutrient concentration and Michaelis–Menton half-saturation constant (Neitsch *et al.*, 2005).

The SWAT considers pollutant reduction through filter strips within each HRU. The reduction rate is estimated as a function of HRU-averaged filter strip width (FILTERW) without considering the spatial distributions of filter strips. SWAT does not simulate dynamic processes of nutrient conversion and pollutant reduction within the RFB. The filter strip function, however, can be used to mimic the attenuation of sediment and nutrient loadings by RFBs with a simplification of the transportation of pollutants from upland fields to receiving waters. SWAT separately simulates trapping efficiencies of a filter strip for surface and subsurface components using FILTERW as a user input for each HRU. The same surface trapping efficiency is used for different constituents, including sediment, organic nitrogen, nitrate nitrogen in runoff, mineral phosphorus sorbed to sediment in surface runoff, soluble phosphorus, and organic phosphorus. Similarly, the same subsurface trapping efficiency is used for NO_3-N removal through lateral flow and groundwater.

MODELING PROCEDURES

This study was conducted in four steps. First, eight subwatershed-boundary and stream-network configurations were delineated based on alternative CSA selection in the AVSWAT-X interface. Second, four different levels of FILTERW, representing alternative RFB widths, were developed for each subwatershed configuration. Third, hydrology and water quality components of SWAT were calibrated based on the watershed configuration most closely resembling the observed drainage density. Fourth, the derived calibration parameter set was applied to all 32 combinations of subwatershed and FILTERW configuration without changes in the parameter set and variability in the output was examined.

WATERSHED SUBDIVISION

Eight different watershed subdivisions were configured by selecting CSA values between 4.0 and 188.1 ha, corresponding to 0.25 and 12% of the total watershed area, respectively (Table 1). The CSA value recommended by AVSWAT-X interface was 31.44 ha, which corresponds to 2% of the total watershed area (subsequently referred to as the 2% CSA). This value was recommended by previous research in northeastern Indiana (Arabi *et al.*, 2006) and Iowa (Jha *et al.*, 2004). HRUs within each subwatershed were created using a 0% threshold setting for both land use and soil class. This threshold setting for land use and soil type allows the average CN value for the entire watershed to be constant over the entire range of CSA values. In order to understand the response of SWAT to the different watershed configurations and RFB representations, percent areas of each land use and soil type within the entire watershed should not be changed for the entire range

TABLE 1. Derived Properties of the Watershed Delineation for the LRN.

	Subwatershed Delineation Levels							
Critical source area (ha)	4.0	7.9	15.7[1]	23.5	31.4	62.7	125.4	188.1
% of total watershed area	0.25	0.50	1.00	1.50	2.00	4.00	8.00	12.00
Number of subwatersheds	180	96	54	35	29	17	11	3
Number of HRUs	2,326	1,826	1,422	1,176	1,074	830	656	331
Average subwatershed area (ha)	8.70	16.32	29.05	44.96	54.25	92.35	142.53	522.18
Drainage density (km/km^2)	2.89	2.19	1.67	1.36	1.25	0.83	0.48	0.32
Stream length at the watershed-outlet subwatershed (m)	306	306	306	306	306	306	306	3035

Notes: HRU, hydrologic response unit; LRN, Little River subwatershed N; SWAT, Soil and Water Assessment Tool.
[1]Used for SWAT calibration.

of CSAs because CN, which is one of the most sensitive parameters, is calculated based on the combination of different land use and soil types. Derived subwatershed properties for LRN are summarized in Table 1 for different CSA levels. Drainage density, defined as the ratio of total stream length (km) to total watershed area (km^2), decreased from 2.89 to 0.32 km/km^2 as CSA increased from 4.0 to 188.1 ha. The actual drainage density of 1.59 km/km^2, which was calculated for this application through a spatial analysis of the digitized stream network, was approximately the same as that derived for a CSA of 1% (15.7 ha). Figure 2 shows changes in stream length at the subwatershed containing the watershed outlet and drainage density as a function of different watershed subdivision levels. The watershed subdivision associated with a 1% CSA was selected for calibration of the hydrology and water quality components of SWAT because the spatial distribution of riparian buffers is most directly related to the stream network, and the drainage density most closely approximated the observed drainage density at this CSA level.

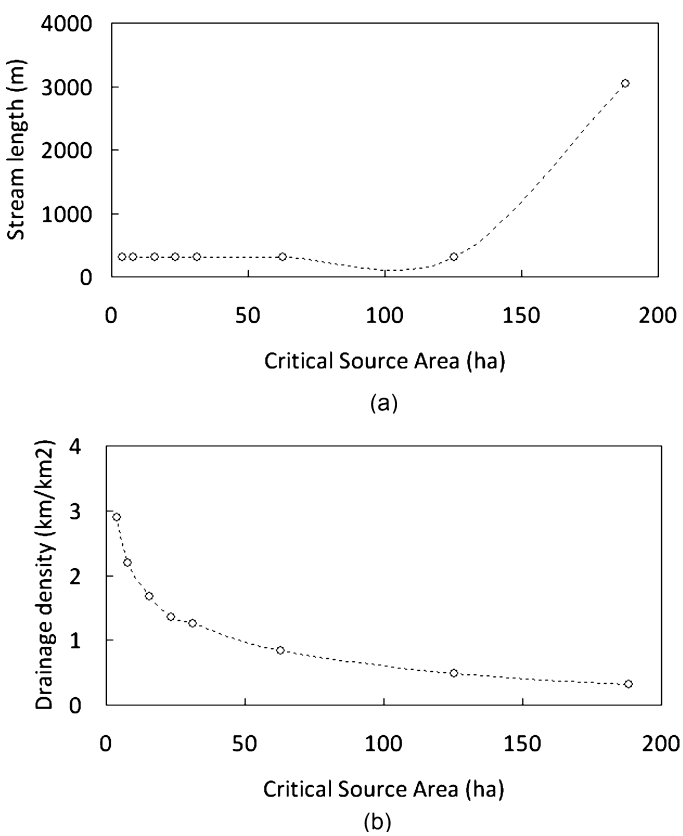

FIGURE 2. Changes in (a) Stream Length at the Subwatershed Which Contains Watershed Outlet and (b) Drainage Density According to Different Watershed Subdivision Levels.

RIPARIAN BUFFER REPRESENTATION WITH FILTERW

The criteria used to select values of FILTERW were based upon the estimated trapping efficiency of filter strips as calculated by SWAT (Neitsch *et al.*, 2005). Trapping efficiencies were divided into three stages: Stage 1 where both surface and subsurface trapping efficiency are less than 100%; Stage 2 where surface trapping efficiency reaches 100% while subsurface trapping efficiency is less than 100%; and Stage 3 where both surface and subsurface trapping efficiencies are 100% (Figure 3). The value of FILTERW delineating Stages 1 and 2 was 30 m while the value separating Stages 2 and 3 was 49 m (Figure 3). Representative widths of 30 and 50 m were selected to represent these breakpoints. An additional FILTERW width of 14 m was selected as it yielded a sediment trapping efficiency of 80%, which is representative of current RFBs in the LREW. A study by Sheridan *et al.* (1999) showed that the measured ranges of total reductions in sediment loads varied from 68 to 95% according to different managements applied. However, management information on RFB areas is not available at the watershed scale and FILTERW in SWAT does not have physical meaning for considering different managements within the buffer areas. As a result, 14 m FILTERW was selected for representing average sediment reduction rate from the study.

Using the GIS, buffers with different widths (14, 30, and 50 m) were created according to the delineated stream network and the land use GIS layer was clipped using the buffers. The fraction of forest area within these buffers was then determined. Examples of different watershed configurations are given in Figure 4. Even though the actual land cover condition within the watershed does not change, the

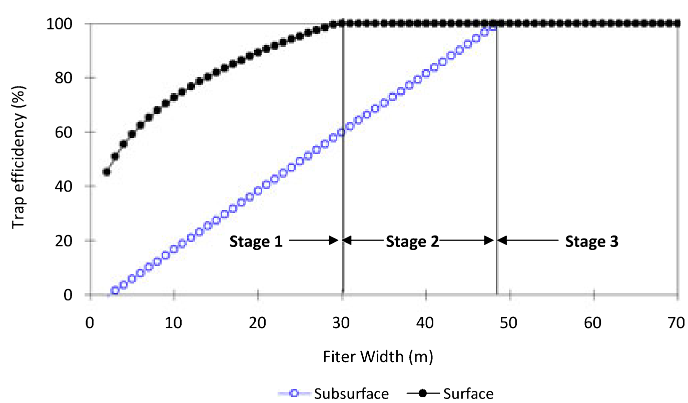

FIGURE 3. Three Divided Stages of FILTERW According to the Surface and Subsurface Trapping Efficiency.

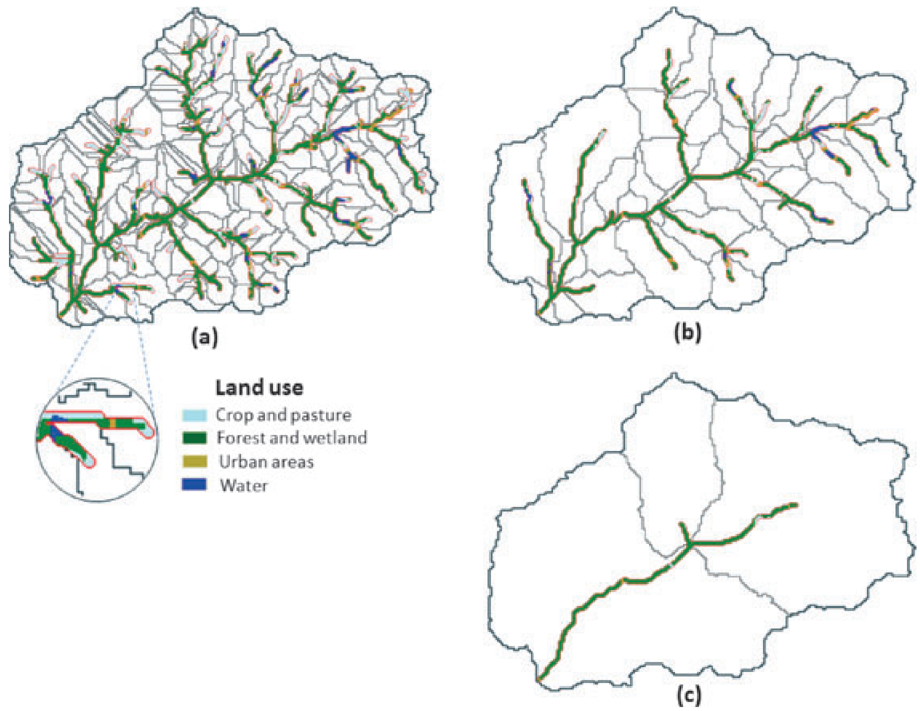

FIGURE 4. Example of Spatial Distribution of Riparian Buffer and Conservation Practices in the Little River Experimental Watershed According to Different Critical Source Areas (CSAs): (a) 4.0 ha CSA, (b) 23.5 ha CSA, and (c) 125.4 ha CSA.

percentage of forest area within buffer zones does change mainly due to differences in the delineated stream networks as a function of the selected buffer width and CSA level (Figure 4, Table 2). Average percent of forest contained within the selected buffer zones increased for all widths examined as CSA increased. As the CSA threshold increases the modeled stream encompasses fewer of the lower order streams, increasing the likelihood that the modeled stream system will be surrounded by wider forested buffers. The percentage forest decreased for all CSAs as the width increased. The FILTERW value for each delineated subbasin was calculated from the forested fraction of the total area of 14, 30, and 50 m buffer areas and varied by subbasin (14 m variable, 30 m variable, and 50 m variable FILTERW representations). The precise width of the buffer for each of the variable FILTERW representations was determined by the width of the buffer that actually contained forest cover. For example, if 80% of a 30 m stream buffer was forested, a 24 m (30 m $\times$ 0.80) FILTERW was used as input for that subbasin. For 30 m variable FILTERW representation, the maximum FILTERW was set at 30 m when all the buffer area consisted of forest, while the minimum of 0 m correspond to no forest within the 30 m width. The computed FILTERW for each subbasin was used to adjust for spatial variation in forest density within each buffer around the stream. In addition to the three variable FILTERW configurations for each subbasin, a

TABLE 2. Average Percent of Forest Area Within the Selected Buffer Widths.

		Subwatershed Delineation Levels							
Critical Source Area (ha)		4.0	7.9	15.7	23.5	31.4[1]	62.7	125.4	188.1
% of total watershed area		0.25	0.50	1.00	1.50	2.00	4.00	8.00	12.00
Average % of forest area within selected stream buffer widths	14 m constant[2]	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0
	14 m variable	69.5	75.4	78.5[3]	81.0	81.6	88.9	94.2	94.1
	30 m variable	66.2	72.6	76.0	78.7	79.3	87.1	93.5	93.9
	50 m variable	59.1	66.9	70.7	73.9	74.6	83.0	91.0	92.3

[1]Recommended CSA value by AVSWAT-X interface.
[2]Areas within the 14 m buffer were assumed to be forest or wetland.
[3]Used for SWAT calibration.

constant FILTERW of 14 m was evaluated for all delineated subbasins (14 m constant FILTERW).

MODEL CALIBRATION

The hydrology and water quality components of SWAT were calibrated using the configuration of 1% (15.7 ha) CSA subdivision level and the 14 m variable FILTERW. No validation period was considered because the main focus of this research was to analyze the response of SWAT to different watershed delineations for both CSA and FILTERW. A three-year equilibration period was followed by a four-year calibration for January 2003 to December 2006. The three-year equilibration period was used to obtain realistic initial hydrology and water quality parameters. The same average rainfall inputs were used for all watershed subdivision levels within LRN. The 14 m width of stream buffer was selected because it yielded the 80% sediment reduction for the buffer that was closest to that observed in previous research on LREW by Sheridan *et al.* (1999). Sensitive parameters influencing upland and in-stream processes selected from earlier LREW SWAT studies were used to calibrate the hydrology and water quality components (Table 3). Sediment yield was also calibrated by considering upland erosion estimated in previous studies (Sheridan *et al.*, 1982; Lowrance *et al.*, 1986). Estimated USLE erosion based on previous

TABLE 3. List of SWAT Parameters Selected for Calibration.

Parameters	Descriptions	Default	Range		Calibrated Value
			Lower	Upper	
ESCO.bsn	Soil evaporation compensation factor	0.95	0	1	0.926
GW_REVAP.gw	Rate of transfer from shallow aquifer to root zone	0.02	0.02	0.2	0.039
GW_DELAY.gw	Time required for water leaving the bottom of the root zone to reach the shallow aquifer (days)	31	0	500	0.036
GWQMN.gw	Threshold water depth in shallow aquifer for return to reach to occur (mm)	0	0	5000	66.7
CN2.mgt	Curve number for crop areas - non-crop	Variable	35	98	
	Crop				−8.2%
	Non-crop				−19.3%
ADJ_PKR.bsn	Peak rate adjustment factor for sediment routing in the subbasin (tributary channels)	1.0	0.5	2.0	1.9
SPEXP.bsn	Exponent parameter for calculating sediment reentrained in channel routing	1.0	1.0	2.0	1.0
SPCON.bsn	Linear parameter for calculating the maximum amount of sediment that can be reentrained during channel sediment routing	0.0001	0.0001	0.01	0.00121
CH_N(2).rte	Manning's n value for the main channel	0.014	0.01	0.3	0.035
FILTERW.mgt	Width of edge-of-field filter strip (m)	0	0	49	14
BIO_E.crop	Radiation-use efficiency or biomass-energy ratio [(kg/ha)/(MJ/m^2)]	Variable			
	Cotton	15	10	90	11
	Peanut	20	10	90	17
EXT_COEF.crop	Light extinction coefficient	Variable	0	1	
	Cotton	0.65	0.5	1.3	0.5
	Peanut	0.65	0.6	0.8	0.6
PSP.bsn	Phosphorus availability index	0.4	0.01	0.7	0.15
AI0.wwq	Ratio of chl a to algal biomass (μg-chl a/mg algae)	50	10	100	87
RHOQ.wwq	Algal respiration rate at 20°C (/day)	0.3	0.05	0.5	0.05
AI1.wwq	Fraction of algal biomass that is nitrogen (mg N/mg algae)	0.08	0.02[1]	0.09	0.06
AI2.wwq	Fraction of algal biomass that is phosphorus (mg P/mg algae)	0.015	0.01	0.02	0.02
RS5.swq	Organic phosphorus settling rate in the reach at 20°C (/day)	0.05	0.001	0.1	0.1
HLIFE_NGW.gw	Half-life of nitrate in the shallow aquifer (days)	365	0	365	1.0
ANION_EXCL.sol	Fraction of porosity (void space) from which anions are excluded	0.5	0.01	1	0.8
ERORGP.hru	Phosphorus enrichment ratio for loading with sediment	0	0	5	5

Note: SWAT, Soil and Water Assessment Tool.
[1]Santhi *et al.* (2006).

monitoring studies ranged from 5.9 (Sheridan *et al.*, 1982) to 7.5 tons/ha (Lowrance *et al.*, 1986). Parameters influencing stream erodibility, including the channel erodibility factor (CH_EROD) and the channel cover factor (CH_COV), were set to zero to eliminate further stream degradation within the current analysis. These parameter settings were determined to be reasonable for LREW due to low streamflow velocities, low channel slopes, and dense vegetation in the watershed (Sheridan *et al.*, 1982). Sediment routing in the channel was calibrated against the sediment yield at the watershed outlet using SPEXP, SPCON, and CH_N2 parameters. It was initially difficult to obtain realistic reduction rates for both sediment and nitrogen using FILTERW. When FILTERW of 14 m was considered to properly estimate the sediment reduction, simulated nitrogen reduction by RFBs was underestimated. As a result, an additional parameter, half-life of nitrate in the shallow aquifer (HLIFE_NGW) was used to generate more realistic nitrogen reduction through the RFBs. The assumption in SWAT, in which the same trapping efficiency is used for different surface-related constituents and both surface and subsurface trapping efficiencies are estimated based on FILTERW, may cause difficulties in considering different RFB reduction rates for different constituents. In addition to ANIO_EXCL, HLIFE_NGW and PSP, which are sensitive to upland nutrient processes, in-stream nutrient transformation processing based on the QUAL2E algorithm was evaluated for adjusting TN and TP yields at the watershed outlet by changing sensitive in-stream parameters such as AI0, RHOQ, AI1, AI2, and RS5 (Table 3).

For model calibration, percent error (PE) was selected as a quantitative measure for comparing observed and simulated total runoff, and sediment and nutrient load for the entire simulation period. The Nash-Sutcliffe Efficiency Index (NSE) (Nash and Sutcliffe, 1970) was selected as a correlation-related statistic for the monthly output. Based on the study by Moriasi *et al.* (2007), model performance was considered as satisfactory, if monthly NSE > 0.50 and PE for streamflow, sediment, and nutrients was within ±25, ±55, and ±70, respectively.

$$\mathrm{PE} = \frac{\sum_{i=1}^{t}(P_i - O_i)}{\sum_{i=1}^{t} O_i} \times 100 \tag{1}$$

$$\mathrm{NSE} = 1 - \frac{\sum_{i=1}^{t}(O_i - P_i)^2}{\sum_{i=1}^{t}(O_i - \overline{O})^2} \tag{2}$$

where PE is percent error of a prediction (%), NSE is Nash-Sutcliffe Efficiency Index, O_i is observed value, $\overline{O}$ is average observed value, P_i is predicted value, and t is the number of observed values.

MODELING OF FILTERW AND WATERSHED SUBDIVISION

The calibrated parameter set was applied to all 32 watershed configurations based on eight watershed subdivision levels (0.25, 0.5, 1, 1.5, 2, 4, 8, and 12% CSAs) and four RFB representations (14 m constant, 14 m variable, 30 m variable, and 50 m variable). Water quality parameters in SWAT can be divided into two groups: parameters involved in computing the transport of pollutants from upland areas to the stream (HRU parameters) and parameters involved in estimating the movement of pollutants within streams (in-stream parameters). The overall water quality response of the watershed is a result of dynamic interactions between the two groups of model components according to changes in both watershed subdivision and spatial representation of RFBs. To separately examine the responses of HRU and in-stream components, total sediment and nutrient yields at the watershed outlet and loads from HRUs were examined from reach output (.rch), and HRU output (.hru), respectively.

CALIBRATION RESULTS

Model performance measures for the calibration period from 2003 to 2006 are shown in Table 4. PE for total streamflow, sediment, TN, and TP were −0.3, −6.9, 4.3, and 8.2%, respectively. Only monthly NSE values for streamflow and TN were within the satisfactory ranges (0.86 and 0.68 monthly NSE values, respectively). Visual comparison indicated good agreement between observed and simulated monthly streamflow and TN loads on LRN. Monthly NSE values for sediment and TP were within the unsatisfactory range for monthly NSE value less than 0.5. Overall, the model performance for hydrology and TN were satisfactory while sediment and TP were unsatisfactory. Among several factors contributing to this, uncertainty of measured data should be considered. Low sediment concentrations in the observed streamflow at LREW are difficult to measure and highly variable (Sheridan and Hubbard, 1987; Hubbard *et al.*, 1990). Possible erosion from

TABLE 4. Model Performance Statistics for the Simulation Period (2003-2006).

	Streamflow (mm)		Sediment (ton)		Total Nitrogen (kg)		Total Phosphorus (kg)	
Measures	**Observed**	**Simulated**	**Observed**	**Simulated**	**Observed**	**Simulated**	**Observed**	**Simulated**
2003	422	395	87	156	6,226	5,340	873	901
2004	437	436	364	280	5,796	6,412	740	1,129
2005	525	481	420	286	7,623	6,861	1,950	1,230
2006	167	235	42	127	2,173	4,148	182	791
Total	1,551	1,547	913	849	21,818	22,761	3,745	4,051
Percent error (%)	−0.3		−6.9		4.3		8.2	
Monthly NSE	0.86		0.41		0.68		0.48	

Note: NSE, Nash-Sutcliffe Efficiency Index.

roads and roadside drainage ditches is also unaccounted for by SWAT. Known model limitations in simulating dynamic nutrient transformations within RFBs may also contribute to the lack of goodness-of-fit (Figure 5d).

IMPACTS OF FILTERW AND WATERSHED SUBDIVISION

Hydrology

Because the FILTERW function does not affect hydrology parameters, the hydrology components of SWAT were not influenced by different RFB representations. Total streamflow fluctuated only slightly in response to watershed subdivision level, with a 0.8% difference between maximum total streamflow (388.7 mm) at 1.5% CSA (23.5 ha) and minimum total streamflow (385.7 mm) at the smallest CSA (4.0 ha). This is largely due to the selection of the 0% threshold for both land use and soil class which yielded a constant CN across the entire range of CSAs. All hydrology components of SWAT output were stable with 1.4, 2.8, and 1.6% differences between maximum and minimum values for surface flow, lateral flow, and baseflow, respectively. This stable response of hydrology components of SWAT to changes in CSA is consistent with results from previous studies (Fitz-Hugh and Mackay, 2000; Jha *et al.*, 2004).

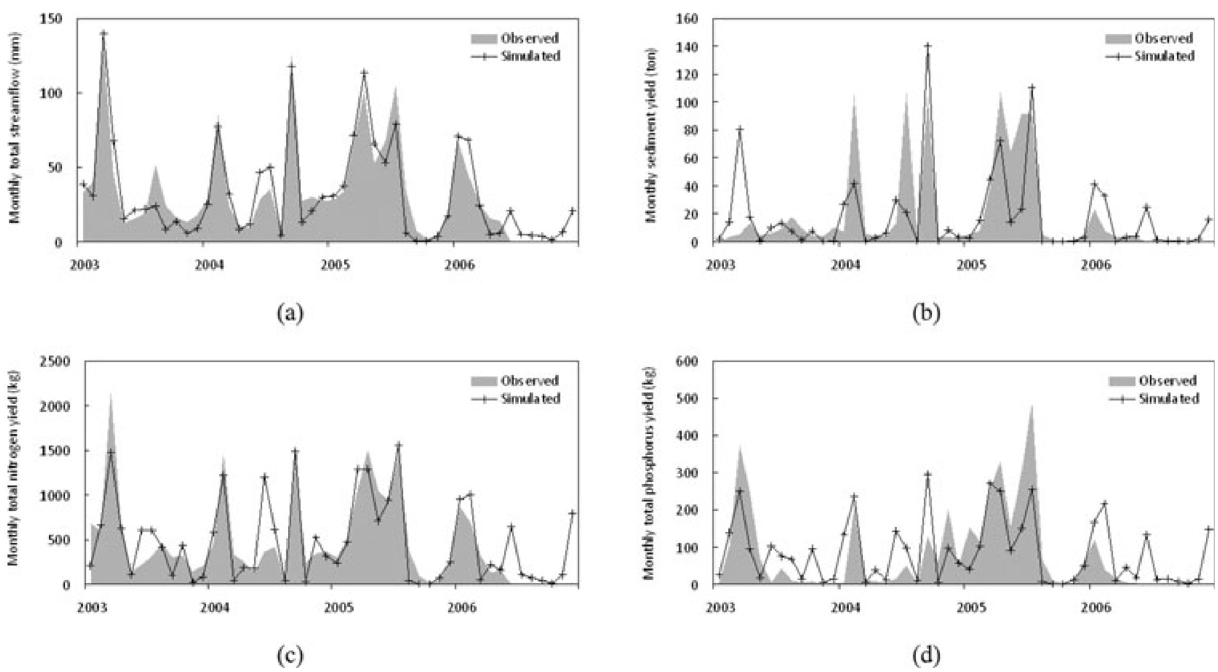

FIGURE 5. Observed and Simulated Monthly and Annual (a) Streamflow, (b) Sediment Yields, (c) Total Nitrogen (TN) Yield, and (d) Total Phosphorus (TP) Yield at the Watershed Outlet.

Sediment

Soil erosion and sediment load from HRUs were evaluated as a function of CSA without considering RFBs because sediment yield at the watershed outlet is a cumulative response to upland erosion and transport (Figure 6). Soil erosion simulated by MUSLE decreased by 3.2% as CSA increased from 4.0 to 188.1 ha, with some fluctuations between the smallest and largest CSAs. Among MUSLE factors, area-weighted average slope factor showed the highest correlation with MUSLE soil erosion (Figure 6a) with a coefficient of determination of 0.52 (data not shown). Sediment load from HRUs and soil erosion showed different trends in response to CSA level (Figure 6). Despite a small decrease in soil erosion (Figure 6a), predicted sediment load from HRUs increased by 19.5% over the full CSA range (Figure 6b). This opposite response can be explained by the nature of the SWAT algorithm that estimates sediment delivery from source areas to streams. Sediment delivery in SWAT is a function of peak runoff from the subwatershed (Neitsch *et al.*, 2005) and peak runoff rate increases as subwatershed size increases. The decrease in sediment erosion with increased CSA value, therefore, was offset by a relatively higher rate of increase in sediment transport from source area to stream.

The effects of combined watershed subdivisions and FILTERW configurations on predicted sediment output are shown in Figure 7. The constant 14 m FILTERW (Figure 7a) and the variable 14 m FILTERW (Figure 7b) configurations responded differently to changes in watershed subdivision for total sediment loads from HRUs (solid lines). Total sediment loads from HRUs increased slightly with CSA level when a constant 14 m filter strip width was simulated (Figure 7a). All stream buffer areas were assumed to be 100% forested in the 14 m constant FILTERW representation. As a result, watershed subdivision was the only variable influencing estimated sediment loads from HRUs for this RFB configuration. Total sediment loads from HRUs decreased with CSA level, however, when a variable FILTERW was simulated (Figures 7b through 7d). The average percentage of forest within these 14, 30, and 50 m stream buffers increased with CSA level (Table 2) because RFBs were more well developed around larger streams, especially near the watershed outlet, compared with the smaller tributaries. Increasing CSA for any variable FILTERW configuration, therefore, increased the total forest buffer area, and decreased the total sediment load. This had the greatest impact for the 50 m variable buffer which yielded the greatest increase in total forest area as a function of increasing CSA. In these variable filter strip representations, the overall response of sediment load from HRUs is a combined result of different watershed subdivisions and FILTERW configurations. The relative effect of increased FILTERW was greater than the relative effect of different watershed subdivisions, with an overall decrease in the sediment load.

Total sediment yields at the watershed outlet for the 14 m constant FILTERW (Figure 7a) and 14 m variable FILTERW (Figure 7b) showed similar trends, regardless of sediment load from HRUs. Within the range of 0.25% CSA (4.0 ha) and 8% CSA (125.4 ha), sediment yield increased with a pattern similar to the sediment load from HRUs with no filter strips as shown in Figure 6b. Simulated sediment yield at the watershed outlet, however, increased rapidly by 62% as CSA increased from 8 to 12% (188.1 ha). This dramatic increase in sediment yields might be attributed to the stream length within the outlet-subwatershed that contains the watershed outlet. Trends in steam length at the outlet-subwatershed and sediment yields at the outlet were similarly correlated with watershed subdivision level (Figure 2a). In general, total sediment yields at the outlet are influenced by the characteristics of channel segments within the outlet-subwatershed. The SWAT model estimates sediment deposition, degradation, and the amount of sediment that is transported out of the channel segment as a function of the volume of water in the stream segment. As a result, sediment

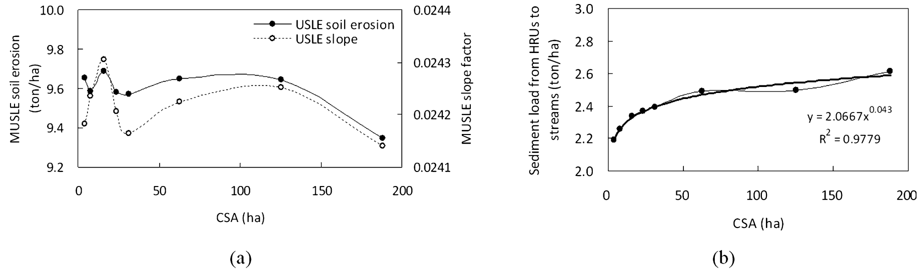

FIGURE 6. Responses of (a) MUSLE Soil Erosion and MUSLE Slope Factor and (b) Sediment Load From HRUs According to Changes in Critical Source Area (CSA) Without Considering Sediment Reductions by the FILTERW Function.

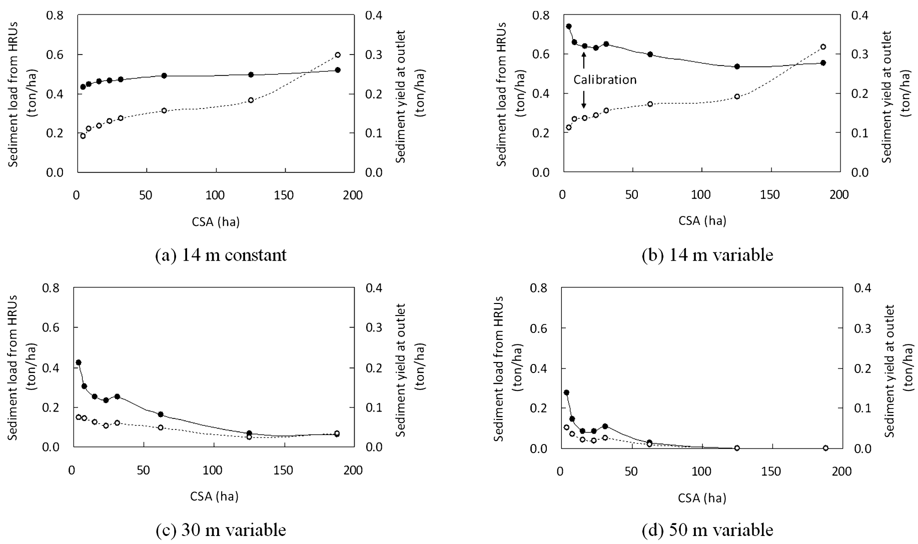

FIGURE 7. Comparison of Annual Sediment Yields at the Watershed Outlet (dashed line) and Annual Sediment Loads From HRUs (solid line) for Different Critical Source Areas (CSAs) and Filter Strip Width (FILTERW), When Stream Erodibility Factors Are Ignored: (a) 14 m Constant FILTERW, (b) Variable FILTERW Based on 14 m Stream Buffer, (c) Variable FILTERW Based on 30 m Stream Buffer, and (d) Variable FILTERW Based on 50 m Stream Buffer.

routing processes can be sensitive to stream length. The large increase in the stream length within the outlet-subwatershed associated with higher values of CSA (Figure 2a) resulted in a significant increase in predicted sediment yield.

Sediment yields for the 30 m variable FILTERW (Figure 7c) and the 50 m variable FILTERW (Figure 7d) responded differently than for the 14 m constant and variable FILTERW representations. Sediment yields at the watershed outlet followed the fluctuations of sediment load from HRUs for the 30 and 50 m FILTERW representations (Figures 7c and 7d). Sediment yields for these FILTERW representations also showed trends similar to each other. As buffer widths increased, the results were less sensitive to changes in CSA.

This differential response of sediment yields at the watershed outlet (14 m constant and variable FILTERW *vs*. 30 and 50 m variable FILTERW) can be explained by a combination of estimated sediment transport capacity and erodibility factors within a stream segment. Sediment routing within a stream segment is determined by comparison of available sediment load and estimated sediment transport capacity. If sediment load from HRUs is greater than transport capacity, sediment deposition will be dominant and sediment load equals transport capacity. Sediment loads from HRUs with 14 m constant and 14 m variable FILTERW representations can be greater than transport capacity because of limited trapping efficiency for these scenarios (Figures 7a and 7b). However, if the sediment loads from HRUs are smaller than the transport capacity, two cases are possible: (1) stream erodibility is high enough to support sediments through channel degradation or (2) stream erodibility is low and sediment yield at the stream outlet is influenced only by available sediment loading from upland areas. In this study, parameters influencing stream erodibility, including channel erodibility factor (CH_EROD) and channel cover factor (CH_COV), were set to zero to eliminate further degradation in a stream. As a result, predicted sediment yields at the watershed outlet were only influenced by decreased sediment loads from HRUs mainly due to increases in trapping efficiency of the filter strips (Figures 7c and 7d).

Nutrients

The effect of different watershed subdivisions and FILTERW configurations on nutrient load from HRUs and yield at the watershed outlet are shown in Figure 8. TN and TP loads from HRUs for the 14 m constant FILTERW (solid lines in Figure 8a) were inversely correlated in their response with increasing CSA. TN load for the 14 m constant FILTERW decreased by 7.3% from 3.90 kg/ha to 3.61 kg/ha over the range of CSA values tested. The dominant constituent of TN loading was NO_3 transported into the main channel in the groundwater (NO3GW), which accounted for more than 80% of total TN. In contrast, TP load increased by 14.9% from 0.53 to 0.61 kg/ha with increasing CSA for the 14 m constant FILTERW. TP trend followed that of sediment loading (Figure 7a) because more than 60% of the TP

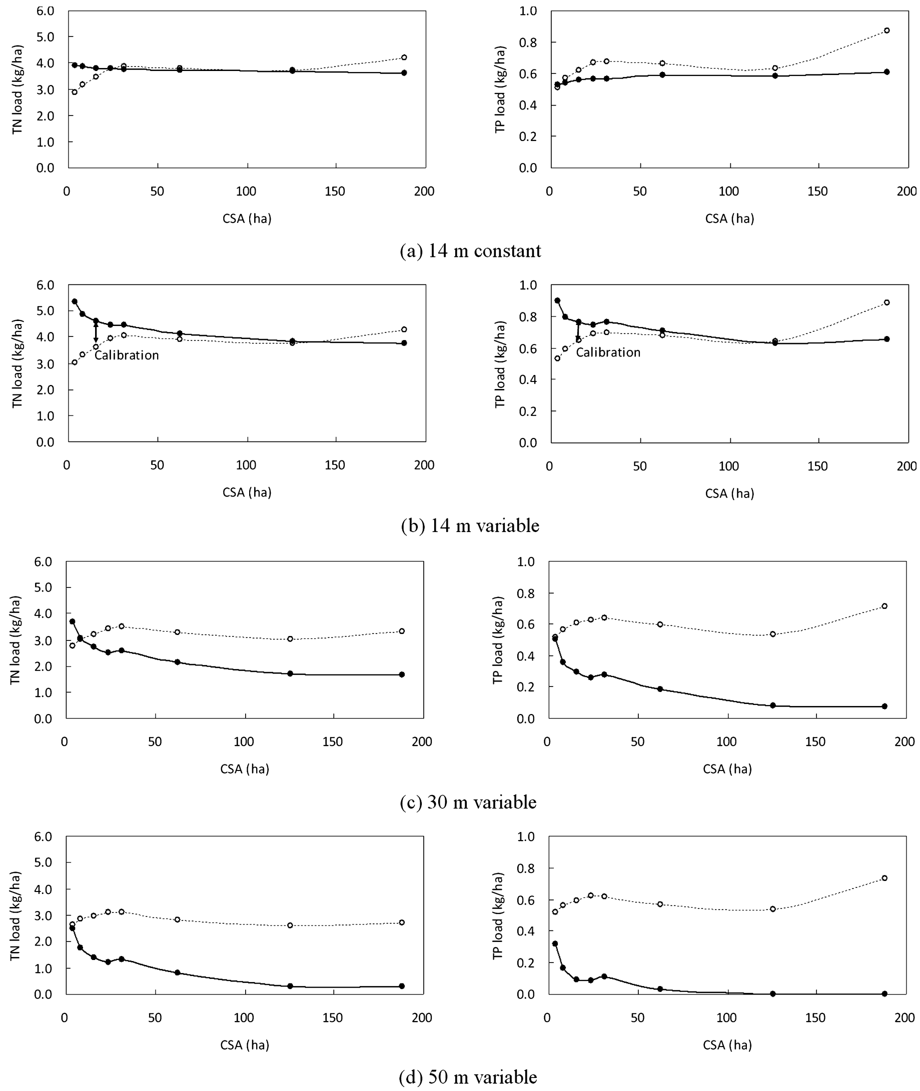

FIGURE 8. Comparison of Total Nitrogen (TN) Loads (left side) and Total Phosphorus (TP) Loads (right side) from HRUs (solid line) and at the Watershed Outlet (dashed line) for Different Critical Source Areas (CSAs) and Filter Strip Widths (FILTERW) When In-stream Nutrient Transformations Are Included: (a) 14 m Constant FILTERW, (b) Variable FILTERW Based on 14 m Stream Buffer, (c) Variable FILTERW Based on 30 m Stream Buffer, and (d) Variable FILTERW Based on 50 m Stream Buffer.

load in the simulation was transported into the stream as mineral phosphorus sorbed to sediment (SEDP) and organic phosphorus which transports with sediment (ORGP). Nutrient loads for both TN and TP decreased with an increase in CSA when the variable FILTERW representations were used (solid lines in Figures 8b through 8d). The nutrient load from HRUs decreased more rapidly at higher variable FILTERW levels due to the more well-developed RFBs around higher order streams within the Little River watershed (Table 2). TN load from HRUs decreased 29.6, 54.8, and 88.7%, respectively, for 14, 30, and 50 m variable FILTERW as CSA increased from smallest (4.0 ha) to largest (188.1 ha). The corresponding values for TP were 26.9, 85.7, and 100.0% for 14, 30, and 50 m variable FILTERW, respectively. Similar to sediment, different FILTERW representations showed greater impacts on the nutrient load from HRUs than different watershed subdivisions.

Total nutrient yields at the watershed outlet for four all FILTERW levels showed similar trends and ranges regardless of nutrient loads entering streams. Nutrient loads increased as CSA increased within the range of 4.0 ha CSA and 23.5 ha CSA. Nutrient loads were stabilized within the CSA range of 23.5 to 125.4 ha and then increased again over the range of 125.4 to 188.1 ha (Figures 8a through 8d). Increases in nutrient loads within the range of 4.0 and 23.5 ha CSAs can be attributed to the methodology by which SWAT calculates in-stream nutrient concentrations based on nutrient loads from HRUs to streams (described in the source code named noqual.f). During the processes of initializing concentration at the beginning and calculating new concentrations at the end of day within the component, nutrient concentrations are set to zero when concentration is less than 0.000001 mg/l. A decrease in average subbasin area and increase in subbasin numbers associated with a decrease in CSA tends to cause a drop in estimated nutrient concentrations. This results in more frequent elimination of low nutrient loads when concentrations drop below the minimum threshold. Additional nitrogen losses can occur during stream nutrient transformations due to settling of organic nutrients and uptake by algae. Nutrient transformations within a stream are determined partially as a function of travel time in each stream segment. For example, change in nitrogen concentration is calculated by multiplying the rate of concentration changes among different nitrogen forms and the travel time in the stream segment. Travel time is computed by dividing the volume of water in the channel by the flow rate. As a result, nutrient transformations within stream can be sensitive to the configuration of the stream network. For a smaller CSA with a dense drainage network (Figure 2b), accumulated travel time within the watershed can be higher than for larger CSA with a less extensive drainage network. Increased travel time appears to have increased the possibility of nutrient removal from the stream network.

Nutrient yields within the 23.5 to 188.1 ha CSA range followed the trend of sediment yield at the watershed outlet in Figures 7a and 7b. The difference between nutrient loads from HRUs (solid line) and nutrient yields at the watershed outlet (dashed line) is equivalent to either a nutrient loss or gain within stream network. When the solid line is higher than dashed line, nutrients are lost in the stream and vice versa. The overall response of nutrient yield at the watershed outlet is a combined result of upland and in-stream processes. As a result, nutrient yield at the watershed outlet is not always smaller than nutrient loads from HRUs when in-stream nutrient transformation was considered. The reason for greater nutrient yields at the watershed outlet than nutrient loads from HRUs can be explained by the algal growth during the nutrient transformation process using the QUAL2E algorithm. Algal growth is simulated when in-stream nutrient transformations are considered. Nutrients can be extracted from the algae when the nutrient concentration is small within a stream segment. As a result, nutrient yields at the watershed outlet were more stable regardless of different FILTERW configurations on upland areas, when in-stream nutrient transformation process was considered.

DISCUSSION

Watershed scale sediment transport, as simulated by SWAT, is the result of complicated interactions among upland sediment load, filtering capacity of buffers, transport capacity of stream, and erodibility of stream. In Coastal Plain watersheds such as LRN, sediment yield at the watershed outlet is controlled by erosion from the upland and the filtering capacity of the buffer because transport capacity and erodibility of the stream channel are limiting (Sheridan *et al.*, 1982). This relationship was represented relatively well by the SWAT FILTERW function. Even though FILTERW in SWAT does not have physical meaning, the relationships between FILTERW and their impact upon sediment loading within this watershed may provide useful guidance for designing optimum riparian buffers within this watershed. For example, the difference in sediment loads from HRUs between 14 and 30 m variable FILTERW (Figures 7b and 7c), which indicate the filtering potential of the increased RFBs, is much greater than the difference in sediment load between 30 and 50 m variable FILTERW (Figures 7c and 7d). There appears to be less of an advantage, therefore, in increasing RFB width within a well-developed RFB area. Such information may also provide useful guidance when developing prioritizations for conservation practices. The potential for reducing pollutant loads from upland areas by additional conservation practices may be greater in areas containing narrow RFBs.

Sediment and nutrient yields at the watershed outlet under different watershed subdivision and FILTERW configurations are significantly influenced by the selected in-stream processes for sediment and nutrients. In order to confirm the in-stream sediment process of SWAT, a different parameter set was developed by considering stream degradation with 0.5 of CH_EROD and 0.5 of CH_COV values just for demonstration purposes. Resulting sediment yields at the watershed outlet and sediment load from HRUs as a function of CSA and FILTERW are shown in Figure 9. Similar to Figure 7, sediment loads from HRUs decreased as the FILTERW increased (Figure 9a). However, sediment yields at the watershed outlet were stable regardless of sediment loads from HRUs (Figure 9b). Unlike the simulations in Figure 7 which had no channel erosion, the sediment transport capacity can be met through channel degradation in spite of insufficient sediment loads from HRUs when in-stream channel erosion is allowed. As a result, sediment yields at the watershed outlet in this simulation were not sensitive to changes in FILTERW (Figure 9b).

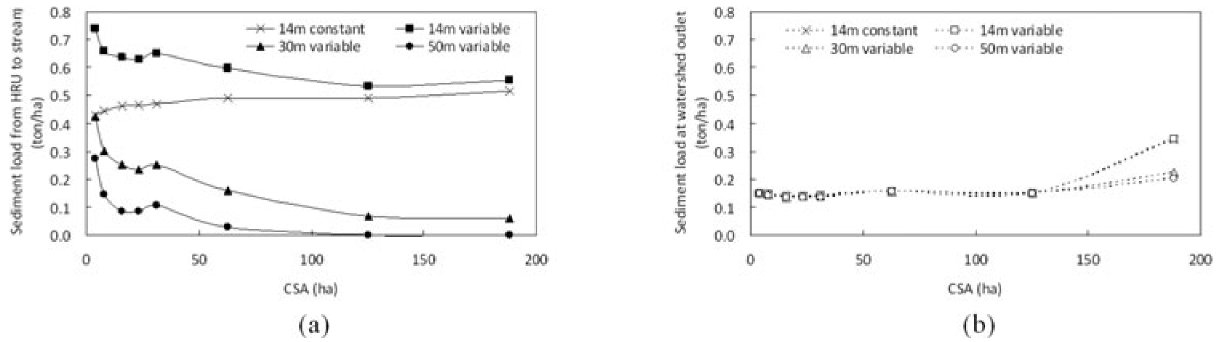

FIGURE 9. Comparison of (a) Sediment Yields at the Watershed Outlet and (b) Sediment Loads from HRUs According to the Changes in Critical Source Areas (CSAs) and Filter Strip Width (FILTERW), When Stream Erodibility Factors Are Considered.

Nutrient yields at the watershed outlet under different CSA configurations can also be influenced by the selected model options for in-stream processing. Figure 10 shows the responses of TN loads from HRUs and yields at the watershed outlet for different CSA and FILTERW representations with no in-stream processing. TN yields at the watershed outlet followed the trend of TN load from HRUs, and TN yields at the outlet were always smaller than TN loads from HRUs regardless of FILTERW configuration.

Different drainage networks can be individually selected by the watershed manager depending on the objectives of the modeling study. If the concern of the modeling is to consider the spatial distribution of land use management within a small watershed, higher resolution of stream network may be necessary. If the concern of the study is to consider lumped impacts of land use management within a regional scale, however, a lower resolution of the stream network may be suitable. For a watershed with extensive RFBs, such as LREW, the degree of watershed subdivision must be considered in conjunction with the spatial distribution of RFBs to properly represent the dominant processes of sediment transportation and deposition. A subdivision level which provides similar drainage density to the actual stream network was used for LREW because the sediment reduction rate by riparian buffers and nutrient yields at the watershed outlet were sensitive to the degree of stream network delineation, especially over the range of smaller CSAs. When an appropriate level of stream network is selected considering the research objectives, the selected stream network can be used as a reference for the watershed delineation procedure in SWAT modeling. Watershed delineation results based on a reference stream network can provide consistent results regardless of different watershed size and users because CSA value based on watershed area may provide different levels of drainage network depending on watershed size. We also suggest that SWAT users provide more detailed information about why the stream network and consequent subwatershed delineations were selected for the modeling purpose.

Selecting an appropriate level of subwatershed delineation is subjective for a watershed but matching

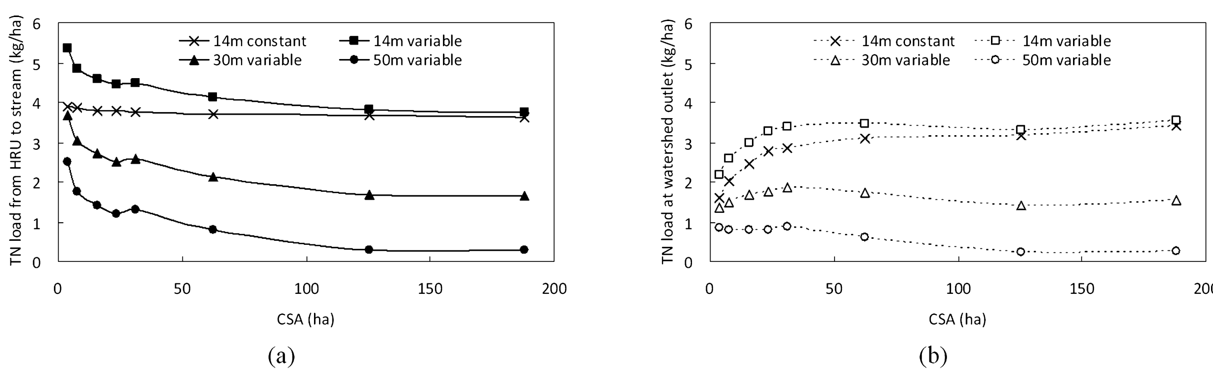

FIGURE 10. Comparison of (a) Total Nitrogen (TN) Loads from HRUs and (b) TN Yields at the Watershed Outlet for Different Critical Source Areas (CSAs) and Filter Strip Widths (FILTERW), When In-stream Nutrient Transformations Processing Was Not Considered.

the stream length generated by selection of CSA and the actual stream length can be used as a guideline. SWAT requires information on the physical characteristics of the main channel within each subbasin to simulate the physical processes affecting sediment transport. This physically based input includes the length of main channel (CH_L), average width (CH_W) and depth (CH_D) of main channel at top of bank, average slope of main channel (CH_S), and Manning's *n* value (CH_N). If sediment yields increase dramatically due to changes in stream length at the outlet-subwatershed, large changes in sediment-related parameters may be required to obtain good calibration results with observed data. In order to avoid resulting inconsistencies in the selection of model parameters for sediment, an upper limit for CSA selection can be established that does not result in a significant increase in stream length within the subwatershed at the watershed outlet. A lower boundary of CSA values can also be used that stabilized modeled nutrient yields at the watershed outlet, which are sensitive to stream network delineations at smaller CSA levels. These threshold CSA limits, however, can vary for different watersheds as a function of hydrologic characteristics and pollutant loadings from upland areas, and must be derived from sensitivity analysis for each watershed.

As a result, CSA ranges which provide stable sediment and nutrient yields at the watershed outlet were suggested for avoiding significant changes in the selected parameter set. For the LRN watershed, the lower boundary of CSA values where the delineated stream network was similar to actual stream network, and where nutrient yields were stabilized at the watershed outlet, was 15.7 ha. The upper CSA boundary where a dramatic increase in sediment yields was initiated was 125.4 ha. Sediment and nutrient yields were stable between these lower and upper boundaries. Sediment, TN, and TP yields between the lower and upper CSA boundaries changed by +41.6, +4.1, and −1.1% at 14 m variable FILTERW, respectively.

SUMMARY AND CONCLUSIONS

In this study, SWAT response to different levels of watershed subdivision and riparian buffer width was evaluated for the Little River experimental watershed N (LRN), Georgia. Eight watershed subdivision levels and four representations of RFB were developed. Watershed subdivision levels included CSAs of 4.0, 7.9, 15.7, 23.5, 31.4, 62.7, 125.4, and 188.1 ha (0.25, 0.5, 1, 1.5, 2, 4, 8, and 12% of the watershed area, respectively). RFB representations included a 14 m constant filter strip width (FILTERW), 14 m variable FILTERW, 30 m variable FILTERW, and 50 m variable FILTERW. The FILTERW component of SWAT affects sediment and nutrients but not streamflow. Hydrology and water quality components of SWAT were calibrated for the watershed configuration at the 1% CSA subdivision and 14 m variable FILTERW level. The calibration parameter set was applied to all 32 watershed configurations. Total sediment and nutrient yields at the watershed outlet and loads from HRUs were compared to separately examine the responses of HRU-related and combined HRU and in-stream SWAT components.

Model performance for hydrology and TN components was satisfactory based upon the criteria of Moriasi *et al.* (2007). PE values for sediment and TP were also determined to be satisfactory while monthly NSE values were unsatisfactory. Unsatisfactory performance of sediment and TP components was partly attributed to the uncertainty of measured data according to measurement condition and techniques and limitation of SWAT in representing the spatial distribution and physical processes of RFBs. Total streamflow was affected very little by watershed subdivision level, varying less than 1% between minimum and maximum total streamflow volumes. Total predicted sediment loads from HRUs increased by 19.5% as CSA increased when no filter strip was considered while total sediment loads from HRUs decreased as CSA increased when the variable filter strip width was used. This was because the larger CSA limited the stream network to larger streams that had a higher percent of forest within the variable filter strip width. Predicted sediment yield at the watershed outlet depended on available sediment, transport capacity, and erodibility within the stream, which are influenced by both watershed subdivision level and RFB representation. When enough sediment was available within the stream through either upland erosion or stream degradation, sediment yield at the watershed outlet was sensitive to the channel length of the stream segment in the subbasin containing the watershed outlet. Nutrient loads from HRUs were not sensitive to the changes in watershed subdivision levels when spatial variation in the filter strips was not considered. Nutrient loads from HRUs, however, decreased with an increase in CSA when the variable filter strip width was used. Sediment, TN, and TP loads from HRUs decreased by 25, 29.6, and 26.9% respectively, as CSA increased from 4.0 to 188.1 ha within the 14 m variable FILTERW configuration. Nutrient yields at the watershed outlet increased with CSA level over the lower CSA range. Similar to the sediment component, nutrient yield at the watershed outlet showed different responses according to the nutrient transformation processes in stream channel. Nutrient yields at the watershed outlet were always smaller than nutrient loads from HRUs and sensitive to the changes in FILTERW when in-stream nutrient processing was not considered. Nutrient yields at the watershed outlet, however, were more stable regardless of FILTERW representation, when in-stream nutrient transformation processing was considered.

A subdivision level which provides a similar drainage density to the actual stream network was suggested as the appropriate CSA value for a specific condition in which realistic representation of spatial distribution of RFBs is important. In general, CSA ranges which provide stable sediment and nutrient yields at the watershed outlet will be useful in order to avoid significant modifications to the parameter set for model calibration. The lower boundary of CSA selection occurs where nutrient yields at the watershed outlet are stabilized; however, this threshold can vary for different watersheds. However, a CSA value which causes changes in stream length within the subwatershed containing the watershed outlet was suggested as the upper boundary of CSA.

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